



The Satoyama Index: A biodiversity indicator for agricultural landscapes

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ABSTRACT

Agricultural development to meet rapidly growing demands for food and biofuel and the abandonment of traditional land use have had major impacts on biodiversity. Habitat diversity is one of the most important factors influencing biodiversity in agricultural landscapes. In this study we propose an ecological index of ecosystem or habitat diversity in agricultural landscapes – the Satoyama Index (SI) – that is discernible under appropriate spatial units (e.g., 6 km × 6 km) from 1 km × 1 km gridded land-cover data available from an open-access web site. A high SI value is an indicator of high habitat diversity, which is characteristic of traditional agricultural systems, including Japanese satoyama landscapes, while a low value indicates a monotonic habitat condition typical of extensive monoculture landscapes. The index correlated well with the spatial patterns of occurrence of a bird of prey (*Butastur indicus*) and species richness of amphibians and damselflies in Japan. The values of the SI also corresponded well to the spatial patterns of typical traditional agricultural landscapes with high conservation value in other countries, for example, the dehesas of the Iberian Peninsula and shade coffee landscapes in Central America. Globally, the pattern of East/South-East Asian paddy belts with their high index values contrasts markedly with the low values of the Eurasian, American, and Australian wheat or corn belts. The SI, which correlates landscapes with biodiversity through potential habitat availability, is highly promising for assessing and monitoring the status of biodiversity irrespective of scale.

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1. Introduction

We are in the midst of the sixth global mass extinction event (McNeely and Scherr, 2002; Thomas et al., 2004). Agriculture is a dominant form of land management globally, and pressure from agricultural development to meet rapidly increasing demands for food and biofuel and the consequent abandonment of traditional land use are recognized as major threats to biodiversity (Fargione et al., 2008; McNeely and Scherr, 2002; Millennium Ecosystem Assessment, 2005; Tilman et al., 2002).

The Convention on Biological Diversity (CBD) provides an international framework for addressing a number of issues regarding the conservation and sustainable use of natural resources and the fair sharing of benefits from genetic resources. Among the CBD's current strategic target is 'to achieve, by 2010, a significant reduction of the current rate of biodiversity loss' (CBD, 2003). One of the major concerns of the parties to the CBD is the development of indicators for monitoring biodiversity in order to assess progress (CBD, 2003).

Monitoring of certain aspects of biodiversity at different levels and scales has been increasing (Billeter et al., 2008; Duelli and Obrist, 2003a; Lindenmayer and Likens, 2009; Muchoney, 2008; Pereira and Daily, 2006; Scholes and Biggs, 2005). A number of indicators have been established to monitor the status and trends of biological diversity and to provide information on improving the effectiveness of policies and management programs on national, regional (Delbaere, 2004), and global scales (Secretariat of the Convention on Biological Diversity, 2010).

The most critical of the issues faced in addressing the targets of the CBD convention are 'biodiversity conservation under agricultural development' (Billeter et al., 2008; Fargione et al., 2008) and 'the sustainable use of natural resources and/or land' (CBD, 2003). Therefore, appropriate indicators for the status of biodiversity and the pressures placed on biodiversity in relation to the intensification of agriculture are needed (CBD, 2003; Firbank et al., 2008). The indicators used in *Global Biodiversity Outlook 3* which summarizes the latest data on status and trends of biodiversity for the future strategy of the Convention on Biological Diversity (Secretariat of the Convention on Biological Diversity, 2010) cover seven focal areas, including reducing the rate of biodiversity loss, addressing the major threats to biodiversity, promoting the sustainable use of natural resources, and maintaining ecosystem integrity.

Habitat diversity and agricultural practices, especially the application rates of fertilizers and pesticides, are considered to be

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the most important factors influencing biodiversity in agricultural landscapes (Benton et al., 2003; Billeter et al., 2008; Duelli, 1997; Firbank et al., 2008; Hendrickx et al., 2007; Hietala-Koivu et al., 2004; Perfecto et al., 2003; Stoate et al., 2009). Agricultural landscapes, especially traditional ones, are a mixture of agricultural and nonagricultural land uses (Harvey, 2007) and comprise a mosaic of land patches differing in the type and intensity of human intervention (Pereira and Daily, 2006). These land patches can include certain types of seminatural habitats, such as grasslands, woodlands, and wetlands, some of which are managed and used for biotic or water resources (McNeely and Scherr, 2002).

The traditional rural landscape common in Japan is the satoyama landscape. The word *satoyama* consists of two Japanese words: *sato*, meaning village or place of human residence with paddies and other croplands, and *yama*, meaning land for exploiting plant resources, such as coppices and grasslands. The most conspicuous ecological feature of the satoyama landscape is the diverse mosaic of agricultural and nonagricultural lands, including woodlands, grasslands, farms, ponds, and creeks, which individually or in a certain combined manner provide a variety of habitats for wildlife and plants (Washitani, 2001). Based on these facts, in 2008 the Japanese government announced at the G8 Environment Ministers Meeting in Kobe, Japan, its 'Satoyama Initiative', which addresses the wise sustainable use of natural resources and land. Similar mosaics of agricultural landscapes, either traditional or newly developed, can be found worldwide and have recently come to be appreciated for their contributions to biodiversity (Daily, 2001; Daily et al., 2003; Harvey et al., 2006; Hendrickx et al., 2007; McNeely and Scherr, 2002; Robinson and Sutherland, 2002; Schulze et al., 2004). In fact, current agricultural–environmental policies, such as some of those stemming from the agri–environmental schemes in Europe, aim to increase landscape heterogeneity to enhance habitat diversity (Benton et al., 2003).

Different types of habitat support different habitat specialists. In Great Britain, species population decline due to agricultural intensification is recognized as particularly marked among habitat specialists, whereas habitat generalists are less affected (Robinson and Sutherland, 2002). Firbank et al. (2008) estimated that habitat diversity accounted for 32% of the variation in plant species richness within Britain's lowland agricultural zone. Moreover, a highly heterogeneous landscape can accommodate multihabitat dwellers, specialists demanding multiple habitats for the completion of their life cycles. In addition a highly heterogeneous agricultural landscape includes many different types of seminatural habitats (Billeter et al., 2008; Daily, 2001; Hietala-Koivu et al., 2004) or secondary forests (Schulze et al., 2004), which have higher potentials for native species diversity.

In this study we propose a biodiversity indicator, which we have called the Satoyama Index (SI), which is a simple composite index of agricultural landscape heterogeneity and the contribution of nonagricultural land use. In this context, the word *satoyama* is used to represent the heterogeneous agricultural landscapes with high conservation values. The ecological basis of this index is the general contribution of β -diversity (between-habitat diversity) (Whittaker, 1960, 1972) to the total biodiversity (Lande, 1996), especially in agricultural landscapes (Hendrickx et al., 2007).

To examine the global pattern of the SI, we calculated the SI with a land-cover data available from an open access web site. We tested the explanatory power of the index at the national scale for Japan, where traditional heterogeneous agricultural land use, which has historically supported various animal and plant species, is rapidly being abandoned. We analyzed the correlation of the spatial pattern of the SI with the pattern of occurrence of the grey-faced buzzard *Buteo indicus* (a bird of prey) and the patterns of species richness of amphibians and damselflies. All of these are representative taxa of satoyama biodiversity; amphibians and damselflies are typ-

ical multihabitat dwellers in agricultural landscapes (Kadoya et al., 2009; Washitani, 2001) and *B. indicus* is a predator of such organisms typical in satoyama landscape, including snakes, amphibians, and insects (Katoh et al., 2009). In addition, we compared the spatial patterns of the SI values of regions in other countries: dehesas landscapes which are ancient agropastoral farming systems (Natura 2000 sites) of the Iberian Peninsula and the shade forest/coffee system in Central America, both of which are known as traditional agricultural landscapes with high conservation values (Daily et al., 2001; Perfecto et al., 2003; Sundseth, 2009).

2. Materials and methods

2.1. Global land-cover data

We chose a 1 km \times 1 km (30") grid size as the elementary spatial unit for calculating the habitat-type diversity index because gridded land-cover data at this resolution is freely available on the World Wide Web (International Steering Committee for Global Mapping, 2009). This gridded land-cover information was created using MODIS data obtained in 2003 (Terra satellite). Each grid cell is assigned one of 20 land-cover categories based on the Land Cover Classification System developed by the Food and Agriculture Organization (Gregorio and Jansen, 2000). The categories are broadleaf evergreen forest; broadleaf deciduous forest; needleleaf evergreen forest; needleleaf deciduous forest; mixed forest; tree, open; shrub; herbaceous; herbaceous with sparse tree/shrub; sparse vegetation; cropland; paddy field; cropland/other vegetation mosaic; mangrove; wetland; bare area, consolidated (gravel, rock); bare area, unconsolidated (sand); urban; snow/ice; water bodies. Although the minimum size of a habitat patch required for population persistence varies largely according to species, with the area of habitat required tending to increase with increasing body size (Peters and Wassenberg, 1983) or other spatial requirements, a spatial scale of 1 km² fulfils the minimum range requirement for a wide range of plants and animals. In fact, the scale is likely to encompass the home range of nesting *B. indicus* (Katoh et al., 2009) and the dispersal distance of amphibians (Van Buskirk, 2005) and damselflies (Purse et al., 2003). We tentatively adopted a basic landscape spatial unit of 6 km \times 6 km consisting of 36 (1 km \times 1 km) elementary spatial units. We assumed that a spatial unit of this size (i.e., 6 km \times 6 km) would be large enough to accommodate persistent populations of the plants and animals that share habitats with humans in the countryside or in satoyama landscapes.

2.2. Calculation of the SI

To determine the value of the SI, we first determined individual basic landscape spatial units containing at least one cell having a land cover classified as 'agricultural' (paddy or cropland; called agricultural landscape units). Next, we calculated landscape heterogeneity among the 36 grid cells comprising each agricultural landscape unit. The agricultural landscape heterogeneity index (ALHI) is calculated using Simpson's diversity index (Lande, 1996), defined as $1 - \sum_{i=1}^S p_i^2$, where S is the number of different land-cover items in a given spatial unit and p_i is the proportion of item i to the 36 elementary grid cells. Grid cells classified as 'urban' were excluded from the calculations. The ALHI ranges from 0 to 1 with higher values indicating greater landscape heterogeneity. Because nonagricultural land use, including natural or seminatural forests, grasslands, and wetlands, is important for biodiversity (Billeter et al., 2008; Daily, 2001; Hietala-Koivu et al., 2004; McNeely and Scherr, 2002; Schulze et al., 2004), we calculated a compound index (the SI) by multiplying the ALHI by the proportion of grid cells classified as 'nonagricultural land except "urban"' within the

basic landscape spatial unit. To standardize the range of values between 0 and 1, the proportion of the nonagricultural land grid cells was multiplied by 36/35. The SI value therefore varies from 0 (homogenous monoculture landscape) to 1 (highly heterogeneous landscape, including a minimum of agricultural cover).

2.3. Comparison with species distribution data in Japan

2.3.1. Species distribution data

Data on the distribution patterns of *B. indicus*, amphibians, and damselflies were obtained from distributional surveys of Japanese wildlife published in 2001, 2002, and 2004, respectively (Ministry of the Environment of Japan, 2009). The national database of wildlife distribution contains the occurrence of species within Japan at a scale of 5'N00" × 7'30"E (about 10 km × 10 km), which is called 'the secondary mesh' and covers all of Japan.

To adjust the spatial resolution of the SI (6 km × 6 km) to conform to that of the secondary mesh, the maximum SI value among all of the 6 km × 6 km spatial units overlapping a 10 km × 10 km secondary mesh grid cell was assigned to that secondary mesh cell. In the following analysis, we targeted the distribution records of Japan's main islands (i.e., records above the 30°N meridian). Of the 4507 grid cells of the secondary mesh, SI values were calculated for 2816 grid cells containing agricultural land use. Surveys on *B. indicus*, amphibians, and damselflies were conducted at least once in 1960, 1530, and 1674 of the 2816 grid cells, respectively.

2.3.2. Statistical analysis

We modelled the occurrence of *B. indicus* and the species richness of amphibians and damselflies in relation to the SI using conditional autoregression (CAR) (Latimer et al., 2006). CAR can consider spatial autoregression in spatial data explicitly, and is therefore appropriate for analyzing our data.

For the occurrence of *B. indicus*, presence/absence data were modelled as

$$\log\left(\frac{p_i}{1-p_i}\right) = \beta_0 + w'_i\beta_1 + \rho_i, \quad (1)$$

and the species richness of amphibians and damselflies was modelled as

$$\log(\mu_i) = \beta_0 + w'_i\beta_1 + \rho_i \quad (2)$$

In Eq. (1), p_i is the probability that the species occurs in cell i , which is assumed to follow a binomial distribution as the ordinal generalized linear model framework; in Eq. (2), species richness was assumed to follow a Poisson distribution with an expected mean of μ_i . w'_i is an explanatory environmental variable associated with cell i (i.e., the SI), β_1 is an associated regression coefficient, and β_0 is an intercept. Each grid cell has an associated random effect ρ_i that adjusts the probability of the presence or species richness of the modelled species dependent upon the values of ρ in cell i 's spatially neighbouring cell j . ρ_i conditioned on ρ_j was a random variable obtained from normal distribution ($N(\cdot)$) specified as follows:

$$\rho_i|\rho_j \approx N\left(\frac{r\sum_{j \in \delta_i} a_{ij}\rho_j}{a_{i+}}, \tau\right), \quad (3)$$

$j \neq i$

where δ_i is neighbouring cells of i , and a_{i+} denotes the total number of cells that are neighbours of cell i ; $a_{ij} = 1$ if sites i and j share the same boundary, and $a_{ij} = 0$ otherwise. We assumed the eight adjacent cells to be neighbours of the focal cell. The variance τ is a hyper-parameter that determines the strength of autocorrelation with neighbours. We also introduced a random variable r ranging from 0 to 1 to enhance the convergence speed of the Markov chain

Monte Carlo (MCMC) iterations. We assigned noninformative priors for the parameters: gamma prior, $\tau \sim Ga(0.1, 0.01)$; for the β s, we assigned normal priors centred at 0 with a large fixed variance of 10; and for r , we assigned a beta prior, $r \sim Be(10, 10)$. This spatially explicit model was fitted by the Bayesian simulation-based method in WinBUGS 1.4 (Lunn et al., 2000), and we obtained posterior distributions of the correlation coefficients β s, which describe the relationships between the spatial distribution of organisms and the SI, independent of spatial autocorrelation. We also calculated the statistical significance of β_1 based on the estimated posterior distribution for each species (groups). The numbers of MCMC steps, burn-ins, and chains were 50,000, 40,000, and 10, respectively, for occurrence of *B. indicus*, and were 10,000, 5000, and 5, respectively, for richness of amphibians and damselflies.

2.4. Comparison with spatial pattern of dehesas and shade coffee landscapes

The dehesas and montados of Spain and Portugal are ancient agropastoral farming systems that strike a delicate balance between productivity and wildlife conservation. These wooded pastures still cover vast areas (50–60,000 km²) of the Iberian Peninsula (Sundseth, 2009). We used Natura 2000 sites (Sundseth, 2009) designated as agricultural landscapes of the peninsula as a measure of dehesa landscapes with high conservation value. Based on the freely available spatial data (European Environment Agency, 2010), we compared the spatial pattern of the sites designated as protected by Natura 2000 with that of the SI values over the Iberian Peninsula, and we also examined the differences the SI values of basic units containing protected areas and the SI values of basic units containing other agricultural areas in the region.

In the shade coffee landscapes in Central America, coffee is grown under a canopy of overstory trees in the traditional 'shade-grown style', which can preserve ecological processes and provide farmers with ecosystem services (Perfecto et al., 2003). Based on information in Blackman et al. (2008), we compared the spatial pattern of the shade-grown coffee sites with that of the SI values for El Salvador as an example where the shade coffee is particularly important ecologically and economically (Blackman et al., 2008). We also examined the differences between the SI values of basic units containing a shade coffee landscape and the SI values of basic units containing other agricultural areas in the region.

3. Results

3.1. Spatial pattern of the SI

Agricultural landscapes are currently the major human-dominated landscape type (Fig. 1a and b), accounting for 35.7% of the basic spatial units comprising the global terrestrial area (excluding Antarctica), and cover large areas over the tropical to temperate zones where climate conditions allow crop cultivation. On a global scale, the values of the SI were calculated as 0.26, 0.00, and 0.64 for the median, 5th, and 95th percentile, respectively. Individual continents or regions each have unique SI patterns, and the SI patterns of South-East Asia (median, 5th, and 95th percentile: 0.39, 0.07, and 0.65) differ significantly with those of the New World (0.27, 0.00, and 0.65; $P < 0.001$; Mann–Whitney U -test) and South Asia (0.06, 0.00, and 0.57; $P < 0.001$; Mann–Whitney U -test). The East Asian to South-East Asian paddy belts are characterized by high SI values, while large tracts with the lowest SI values (indicating extensive monoculture conditions) cover the wheat or corn belts of North America as well as South America, South Asia, Australia, and Africa and the paddy belts of India.

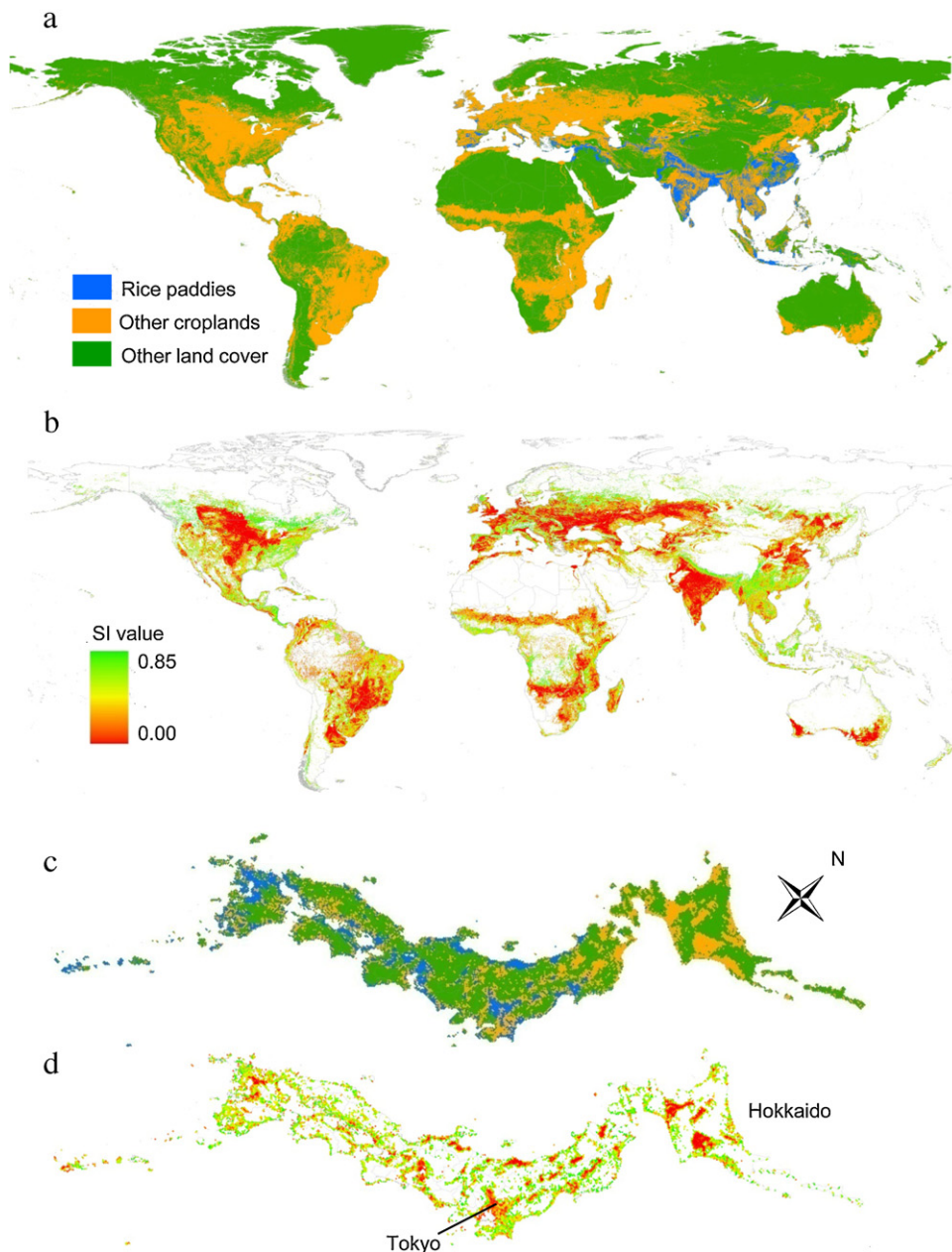


Fig. 1. Spatial patterns of agricultural land cover and the Satoyama Index (SI) at global (a and b) and national (Japan; c and d) scales. The SI was calculated using Simpson's diversity index based on land-cover data in a $6\text{ km} \times 6\text{ km}$ spatial unit; only spatial units that included rice paddies and croplands were included. Each spatial unit consists of thirty-six $1\text{ km} \times 1\text{ km}$ grid cells to which one of 20 land-cover categories is assigned (see text for details). 'Rice paddy' represents a $6\text{ km} \times 6\text{ km}$ spatial unit containing at least one $1\text{ km} \times 1\text{ km}$ grid cell classified as 'paddy field'. 'Other cropland' represents a $6\text{ km} \times 6\text{ km}$ spatial unit that contains at least one $1\text{ km} \times 1\text{ km}$ grid cell classified as 'cropland' or 'cropland/other vegetation mosaic' but none classified as 'paddy field'.

In Japan, agricultural landscape units are distributed along coastal zones and scattered over inland basins (Fig. 1c and d). Compared with the global pattern, Japan has significantly higher SI values (median, 5, and 95 percentile: 0.41, 0.01, 0.65; $P < 0.001$; Mann–Whitney U -test). A large proportion of Japan's mountainous area is not assigned to agricultural landscape units, presumably because individual paddy or cropland patches, if present, are too small to allow grid cells to be classified as 'paddy field' or 'cropland'. The SI values are above 0.4 in more than half of the agricultural landscapes units. The lowest SI values (indicating monoculture conditions; Fig. 1d) are found in some lowland areas, including the Hokkaido agricultural zone where major agricultural development started in the Meiji Era (1868–1921) with the importation of modern farming technologies from Western countries.

3.2. Validation with Japanese data

The scatter diagrams (Fig. 2) show the upper limiting pattern of data by the SI value and suggest that the potential (maximum) species richness of an individual taxonomic group correlates strongly with the SI value. There was a significant positive correlation between the SI value and the occurrence of *B. indicus* ($P < 0.01$; Fig. 2a), the species richness of amphibians ($P < 0.001$; Fig. 2b; 17 species of toads and frogs, one newt, and one salamander), and the species richness of damselflies ($P < 0.01$; Fig. 2c; 39 species).

3.3. Validation with dehesa and shade coffee landscapes

Spatial patterns of the dehesas of the Iberian Peninsula and shade coffee areas of El Salvador corresponded well to high val-

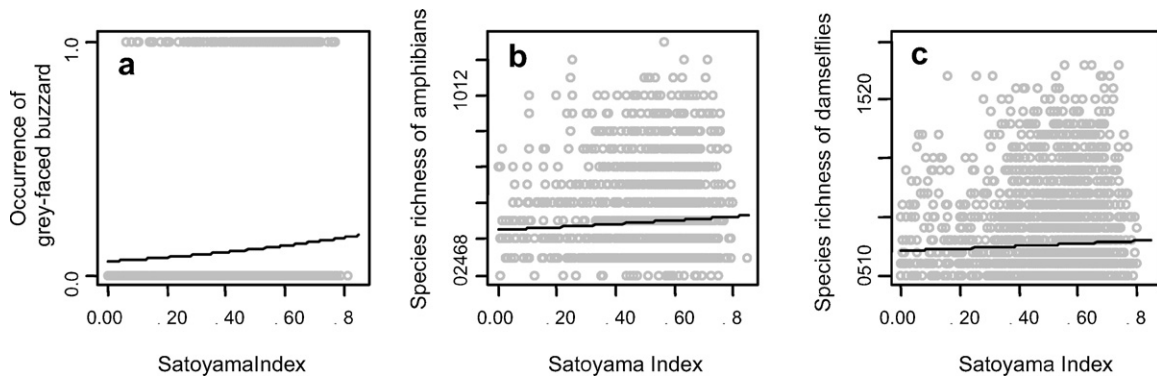


Fig. 2. Relationships between the SI and occurrence of the grey-faced buzzard *Butastur indicus* (a), species richness of amphibians (b), and species richness of damselflies (c) in the Japanese Archipelago. Spatial regression showed significant positive relations as indicated by regression lines: buzzard $y = 1 / \{1 + \exp[-(1.39x - 2.77)]\}$, $P < 0.01$; amphibians $y = \exp(0.328x + 0.921)$, $P < 0.001$; damselflies $y = \exp(0.429x + 0.735)$, $P < 0.01$.

ues of the SI (Fig. 3a and b). There was a significant difference in the SI value between the basic units of Natura 2000 protected sites (median, 5th, and 95th percentile: 0.31, 0.00, and 0.65, respectively) and the other agricultural areas (0.15, 0.00, and 0.58) of the Iberian Peninsula ($P < 0.001$; Mann–Whitney U -test) and between those with shade-grown coffee areas (0.46, 0.23, and 0.62) and other agricultural landscapes (0.15, 0.00, and 0.58) in El Salvador ($P < 0.001$; Mann–Whitney U -test).

4. Discussion

4.1. Agricultural land use and the SI

Our results show that the SI is a good predictor of *Satoyama* biodiversity at the national scale of Japan and correlates well with the pattern of global land use; East/South-East Asian paddy belts have high index values, which contrast markedly with the low values of the Eurasian, American, and Australian wheat or corn belts. SI values vary greatly over Europe where there is a rich diversity of agricultural landscapes that differ in types of land use and in the abundance and pattern of seminatural elements (Billeter et al.,

2008). The Iberian Peninsula, Scotland, and Eastern Europe have relatively high SI values (Fig. 1b). The green zones covering the Iberian Peninsula seem to correspond to the dehesas, a unique traditional silvi-pastoral system (McNeely and Scherr, 2002). In fact, the SI pattern for the Iberian Peninsula corresponded well to the areas of Natura 2000 (European Environment Agency, 2010; Fig. 3), and it was demonstrated that the areas of Natura 2000 had significantly higher SI values than remaining areas of the peninsula (Spain and Portugal).

The SI pattern of the tropical/subtropical regions of South-East Asia is markedly different from the SI pattern found in the tropical/subtropical regions of South America where conversion of tropical forests into agricultural lands is one of the major threats to biodiversity (Daily et al., 2003; Lawton et al., 1998; McNeely and Scherr, 2002; Schulze et al., 2004). There are contrasting patterns also between South and Central America, with higher proportions of ‘greener’ spatial units found in Central America. Shade coffee landscapes would be responsible for the high SI value, at least in part. In fact, in El Salvador, the shade-grown coffee areas had a significantly higher SI values than other agricultural areas. If higher SI values can be attributed to higher proportions of intact remnant or secondary forests and if metapopulation capacities (Hanski and

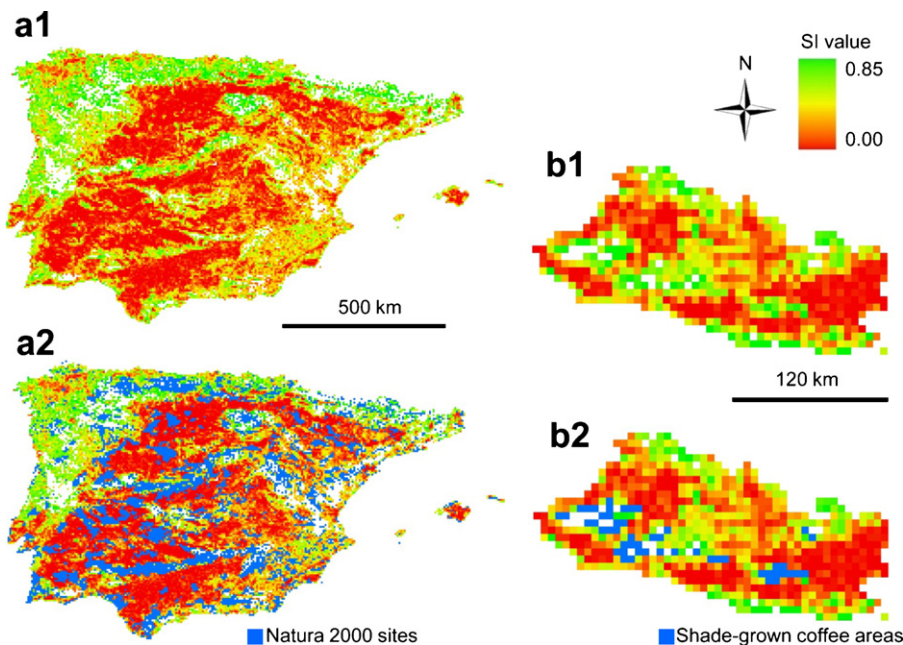


Fig. 3. Spatial patterns of the Satoyama Index (SI) in the Iberian Peninsula (a1 and a2) and El Salvador (b1 and b2). Blue grid cells show the areas of Natura 2000 sites (a2) and shade-grown coffee areas (b2). Scale bars represent 500 km and 120 km for the Iberian Peninsula and El Salvador, respectively.

Ovaskainen, 2000) of at least some of the original flora and fauna species before agricultural development are maintained, threats caused by land conversion may be alleviated to some degree.

4.2. Merits of the SI as a biodiversity index

Establishing operational indicators for the status of biodiversity (Duelli and Obrist, 2003b; Scholes and Biggs, 2005) is difficult because biodiversity is a complex, multiscale, multifaceted entity (Noss, 1990). A species indicator approach is likely to be the most explicit for indicating the status of biodiversity, but opinion on the usefulness of surrogate taxa has been somewhat contradictory (Lawton et al., 1998; Pereira and Cooper, 2006; Sauberer et al., 2004). Different taxonomic groups or guilds behave differently in response to changes in land-use systems (Perfecto et al., 2003; Schulze et al., 2004) or to other drivers (Robinson and Sutherland, 2002). Attempts to find species indicators of agricultural landscapes that are useful in predicting the overall species richness of other taxa have not been successful (Billeter et al., 2008; Sauberer et al., 2004). Large-scale datasets obtained using the same standard method are, for the most part, lacking, and therefore the indicator species approach with existing data is impractical on a global scale, except for a few taxonomic groups, such as vascular plants (Kreft and Jetz, 2007) and birds (Davies et al., 2007).

The SI, which correlates landscapes and the states of biodiversity through potential habitat availability is a highly promising tool for assessing and monitoring the status of biodiversity, irrespective of the scale. Although examining the SI on a global scale is a task for future research, measures relating to habitat diversity are reported to drive global species richness patterns for several taxa (Davies et al., 2007; Guegan et al., 1998; Kreft and Jetz, 2007); thus the SI is also likely to work well in explaining the richness patterns of agricultural landscapes on a global scale. Theoretically, spatial patterns in mosaic (patchy) landscapes are expected to have a strong influence on metapopulation dynamics, especially through the amount of suitable patches and the dispersal of individuals at the landscape level, both of which govern the long-term persistence and abundance of a species population (Hanski and Ovaskainen, 2000). An index that reflects the amount and connectivity of potential habitat should, therefore, provide a more reliable long-term expectation of persistence of populations or communities than an actual distribution that can only be sampled incompletely in time and space.

A possible improvement to the SI could be to add a weighting factor for the various categories of land usage. Each of the 20 land-use categories has a range of habitat biodiversity. For example, the broadleaf deciduous category includes monocultures of *Salix* or *Populus* in western and northern North America and the very diverse *Quercus* forests of eastern North America. Combining all these forests as one category may be concealing the complexity and biodiversity of this land form. Such a revision of land-use categories is beyond the scope of the present study, which describes a simple and general concept for an index that can be projected on a global scale. However, land-use category revision may be required to apply the index to finer scales.

A pan-European study on biodiversity indicators in agricultural landscapes demonstrated that species richness at the landscape scale can be predicted by few variables representing land-use intensity and the spatial structure of both agricultural and seminatural areas (Billeter et al., 2008). A universal consequence of agricultural intensification is the replacement of heterogeneity in habitat structure with the homogeneity of an extensive monoculture of annual crops (Benton et al., 2003; Daily et al., 2003). Agricultural sustainability is a major theme of the United Nation's Millennium Development Goals (Millennium Ecosystem Assessment, 2005). The eco-agricultural approach advocated by McNeely and Scherr (2002) and Scherr and McNeely (2007),

encompasses such areas as agroecology (Altieri, 1985), permaculture (Mollison, 1990), conservation agriculture (Food and Agriculture Organization, 2001), agroforestry, organic agriculture (International Federation of Organic Agriculture Movements, 2000), and sustainable agriculture. This approach also focuses on the management of mosaics with natural areas and other landscape features in addition to biodiversity-friendly agricultural production systems and practices within farmlands.

The SI is a scientifically sound index that can be easily calculated from satellite-based land-cover data and enables comparisons at various scales from local to global according to the grid sizes of the available land-cover data. The SI approach would have the potential to support current efforts, e.g., by DIVERSITAS, to develop improved biodiversity monitoring schemes through the Group on Earth Biodiversity Observation Network programme (2008) or in the context of the Satoyama Initiative. Not only does the SI fulfil the criteria required for a biological diversity index (CBD, 2003), the SI or a similar index may be used in a wider context in environmental impact assessments and life-cycle assessments on goods or services (Schenck, 2001) where appropriate consideration of changes in land use is needed to evaluate the impacts of human actions.

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